Responses of Arctic Grayling (*Thymallus arcticus*) to Acute and Prolonged Exposure to Yukon Placer Mining Sediment

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Underyearling Arctic grayling (*Thymallus arcticus*) from the Yukon River system were expsed for 4 d to suspensions of fine inorganic (≤250 g·L^{-1}) and organic (≤50 g·L^{-1}) sediment and for 6 wk to inorganic sediment (≤1000 mg·L^{-1}) under laboratory conditions. The test sediments were collected from an active placer mining area near Mayo, Yukon Territory. The exposures evoked sublethal responses but did not cause gill damage. Mortalities (10 and 20%) occurred only in experiments at 5°C with inorganic sediment concentrations ≥20 g·L^{-1}. Six weeks of exposure to sediment concentrations >100 mg·L^{-1} impaired feeding activity, reduced growth rates, caused downstream displacement, colour changes, and decreased resistance to the reference toxicant pentachlorophenol, but did not impair respiratory capabilities. Stress responses (elevated and/or more varied blood sugar levels, depressed leucocrit values) were recorded after short exposure (1–4 d) to organic sediment concentrations as low as 50 mg·L^{-1}. Inorganic sediment strengths ≥10 g·L^{-1} caused fish to surface. The lethal and sublethal responses of Arctic grayling to pentachlorophenol were similar to those determined for other healthy salmonid fishes.

De jeunes ombres de l'Arctique (*Thymallus arcticus*) de moins d'un an provenant du bassin du fleuve Yukon ont été exposés en laboratoire pendant 4 j à de fines matières inorganiques (≤250 g·L^{-1}) et organiques (≤50 g·L^{-1}) et pendant 6 sem à des matières inorganiques (≤1 000 mg·L^{-1}). Les matières en suspension utilisées provenaient d'une zone d'exploitation de placers en activité des environs de Mayo, Territoire du Yukon. Les expositions ont provoqué des réponses sublétales mais n'ont pas causé de lésions aux branchies. Il n'y a eu mortalité (10 et 20 %) qu'au cours des essais portant sur des concentrations de matières inorganiques égales ou supérieures à 20 g·L^{-1} effectuées à 5°C. L'exposition pendant six semaines à des concentrations de matières en suspension supérieures à 100 mg·L^{-1} s'est traduite par une baisse de l'alimentation et du taux de croissance et a provoqué un déplacement vers l'aval, un changement des couleurs et une baisse de la résistance au produit toxique témoin, le pentachlorophénol, mais n'a pas eu d'effet sur la capacité respiratoire. Des réactions au stress (teneur en sucre du sang plus élevée ou variable, dépression du leucocrite) ont été notées après les courtes expositions (1 à 4 j) à des concentrations de matières organiques aussi faibles que 50 mg·L^{-1}. Les poissons réagissaient aux teneurs en matières inorganiques de 10 g·L^{-1} ou plus en remontant à la surface. Les réactions létales et sublétales au pentachlorophénol des ombres de l'Arctique étaient semblables à celles notées chez d'autres salmonidés en bonne santé.

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Placer mining in the Yukon is considered to be a viable industry with an historical background (Anonymous 1983). The need to manage fisheries resources and the disruption of fish habitat during the extraction of gold by placer mining presents a potential conflict. While the impact of current and past mining activities on fish and their habitat is not clearly understood, a number of studies have shown site-specific evidence of damage to aquatic organisms and their habitat (Mathers et al. 1981; Singleton et al. 1981; Weagle 1982; Simmons 1984). Adverse effects have generally been attributed to suspended and deposited sediments in receiving waters downstream of placer mining activities (Weir 1979; Mathers et al. 1981; Singleton et al. 1981; Weagle 1982; Simmons 1984; W. Knapp, Department of Fisheries and Oceans, 1090 West Pender Street, Vancouver, B.C. unpubl. data; S. Meyer and R. C. Kavanaugh, U.S. Department of the Inte-

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Earlier studies (Herbert and Merkens 1961; Anonymous 1965; Neumann et al. 1975; O'Connor et al. 1977; Noggle 1978; Sigler et al. 1984; Lloyd 1985) have revealed that sediment suspended in water can cause acute lethal or sublethal effects on fish. Although some nonsalmonid fish species have survived short-term exposures to suspended sediment concentrations as high as 100 $g \cdot L^{-1}$ (Wallen 1951), bioassays carried out by Noggle (1978) indicated a seasonal variability in tolerance of salmonids to natural stream sediment, with concentrations as low as 1200 $mg \cdot L^{-1}$ killing underyearling salmonids. Reduced survival, impaired growth, reduced feeding activity, and lower condition factors of salmonids have been recorded after their exposure to suspended sediment fines as low as 300 $mg \cdot L^{-1}$ (Noggle 1978; Herbert and Merkens 1961; Sigler et al. 1984).

Our laboratory investigations were designed to provide information on the acute and chronic effects of placer mining sediments on Arctic grayling (*Thymallus arcticus*), which are present in many northern mining areas (e.g. Alaska, British Columbia, Yukon Territory). This paper summarizes the results of these investigations (McLeay et al. 1983, 1984) and, to our knowledge, provides the first data on the effects of placer mining sediment on *T. arcticus* under controlled conditions. The results of acute lethal and sublethal bioassay tests, developed previously for evaluating the short-term impact toward salmonids of a variety of aquatic contaminants and other environmental stressors (McLeay and Gordon 1980; Wedemeyer and McLeay 1981), are summarized. Also described are the effects of suspended placer mining sediment ($\leq$1000 $mg \cdot L^{-1}$) on the feeding, behaviour, growth, biological condition, and performance capabilities of underyearling Arctic grayling held in artificial streams for a 6-wk period.

Methods

Underyearling grayling (0.03–1.0 g) were collected by pole seine from the Yukon River drainage basin in July and August 1982 and June 1983. Groups of fish (50–250) were transported by air to the laboratory in Vancouver, British Columbia, using ice-cooled plastic bags containing creek water and an oxygen atmosphere. These fish were held for a minimum of 3 wk before experimentation in outdoor fibreglass aquaria supplied with dechlorinated Vancouver City tap water. The minimum water exchange rate was 2 $L \cdot g$ $fish^{-1} \cdot d^{-1}$ and the fish loading density was <2.5 $g \cdot L^{-1}$ to prevent overcrowding (Sprague 1973).

Fish were fed a variety of food. As there were difficulties in getting young fish (swimup fry) to feed, they were given Biodiet No. 1 (<0.6 mm crumb size; Bioproducts Inc., Warrenton, OR) together with live brine shrimp, beef heart, tubificid worms, *Daphnia pulex*, Oregon Moist Pellet mash, and canned salmon. Later, the fish were fed Biodiet No. 2 followed by Oregon Moist Pellets (1.6 mm), brine shrimp, and sockeye salmon (*Oncorhynchus nerka*) eggs.

Inorganic ("paydirt") and organic ("overburden") sediment samples were collected from an active placer mine along Highet Creek, Mayo, Yukon Territory. The inorganic sediment was screened and pulverized and the organic sediment coarse-screened prior to determination of particle size and shape, moisture content, volatile and fixed residue, rate of oxygen uptake, and major trace inorganic components (McLeay et al. 1983). Particle size analysis of the sediments used in the acute bioassay tests indicated that over 90% of the particles in the samples of inorganic sediment were <0.2 mm, with approximately 70% of particles <50 μm and 60–65% <38 μm (silt and clay), whereas approximately 50% of the organic sediment was >0.2 mm and only 3% was <38 μm. The inorganic sediment used in the prolonged exposure study comprised approximately 70% silt and clay (<38 μm), with 98% <75 μm. The volatile (organic) content of these inorganic and organic sediment samples was 4–5 and 96%, respectively. Additional characteristics of the test sediments are provided elsewhere (McLeay et al. 1983, 1984). In general terms, the test sediments used in these bioassays were characteristic of organic overburden found overlying inorganic "paydirt" and of inorganic fines carried into downstream waters during sluicing operations (Anonymous 1981).

The interrelationship between nonfilterable residue (suspended solids), total residue, and turbidity values for suspensions of each sediment type in freshwater was examined (McLeay et al. 1983). Total residue values determined for suspensions of inorganic and organic sediment sampled during the bioassay tests proved to be good approximations of their total nonfilterable residue (NFR) content.

Acute Exposure Studies

Acrylic 50-L tanks with conical bottoms (Noggle 1978) were constructed for use in the acute survival bioassays, temperature tolerance tests, and stress bioassays. Rectangular baskets of soft mesh nylon netting in a stainless steel frame fitted within the main body of each tank contained the fish and facilitated periodic observations.

Test suspensions were maintained by the continuous recirculation (10.3 ± 0.3 $L \cdot min^{-1}$) of water collected from the conical bottom of each tank. A 100-mL aliquot of each test suspension was taken at the termination of each acute survival test to determine the total residue content to which fish were exposed. Preliminary tests also determined total residue concentrations and patterns of dispersal within the recycle tanks for a range of concentrations of suspended sediment (McLeay et al. 1983).

All acute exposure studies with inorganic or organic sediment were performed with controlled temperature rooms isolated from general laboratory disturbances. Overhead incandescent illumination regulated by photocell and automated rheostat (for provision of 0.5-h "dawn/dusk" of variable intensity) simulated a natural photoperiod.

Fish survival and condition

Groups of 10 fish acclimated to 15 or 5°C for periods of 11 (15°C) or 9 (5°C) wk were placed randomly in a series of 50-L volumes of inorganic (0.05–250 $g \cdot L^{-1}$) or organic (0.05–50 $g \cdot L^{-1}$) sediment in suspension. Water temperature (5 or 15°C, depending upon acclimation temperature), pH, dissolved oxygen content (milligrams per litre), and conductance (microsiemens per centimetre) values for each suspension were monitored daily for 4 d (96 h), together with fish survival. One group of fish acclimated to 15°C was held in 50 g inorganic $sediment \cdot L^{-1}$ for 16 d at this temperature, during which time water quality conditions and fish survival were monitored daily.

All dead fish and those surviving the acute exposure bioassays were inspected for gross lesions and overt signs of disease. Gill tissue from selected groups of fish surviving a 96-h exposure to a range of concentrations of inorganic or organic sediment was removed, fixed (Bouin's), paraffin-embedded, sectioned (6 μm), and stained (hematoxylin/eosin) for subsequent histological examination. Blood samples from these fish were collected in heparinized microhematocrit tubes and analysed for hematocrit values.

Acute stress bioassays

Controlled bioassays were performed to determine the concentrations of inorganic and organic sediment suspensions which were acutely stressful to *T. arcticus*. Basic test procedures were those proven effective for determining threshold strengths of a variety of aquatic contaminants which cause stress responses (elevated blood sugar levels, decreased numbers of circulating leucocytes) in other salmonid fish species (McLeay 1977; McLeay and Gordon 1977, 1979, 1980).

Groups of 10 fish acclimated for 48 h to the recycle tanks in the absence of sediment were subsequently exposed to a range of concentrations of suspended inorganic or organic sediment. Duplicate control groups (freshwater only) were included for each bioassay series. Each group of 20 fish was sacrificed for blood sugar, leucocrit, and hematocrit determinations after a 24-h (15°C-acclimated fish) or 96-h (5°C-acclimated fish) exposure to sediment, using standardized test procedures (McLeay et al. 1983).

Tolerance to hypoxia

Sealed jar (residual oxygen) bioassays were conducted with groups of grayling acclimated to 15 or 5°C and exposed to a range of concentrations of suspended inorganic (15 and 5°C) or organic (15°C) sediment. Basic test procedures were those developed for use with kraft pulpmill effluents (McLeay 1976; Gordon and McLeay 1977) and applied subsequently with other aquatic contaminants (McLeay and Gordon 1980). Ten replicate jars were prepared for each sediment concentration. Time to death and residual dissolved oxygen concentration at death of each fish (one per jar) were determined and compared with values for control fish (McLeay et al. 1983).

Temperature tolerance

The effect of acute sediment exposure on the critical thermal maxima (upper lethal temperature tolerance) for Arctic grayling acclimated to 15 or 5°C and held in a range of concentrations of inorganic or organic sediment was determined. Test procedures, whereby groups of 10 fish in differing sediment strengths were subjected until death to progressively rising temperatures ($1°C \cdot h^{-1}$) from their acclimation temperature, were those used for other aquatic contaminants (McLeay and Howard 1977; McLeay and Gordon 1980).

Reference toxicant tests

Tolerance to the reference toxicant pentachlorophenol (Davis and Hoos 1975) was assessed for the stocks of Arctic grayling used in the acute exposure studies. Acute lethal tolerance to pentachlorophenol was determined for grayling acclimated to 15 and 5°C in the absence of any sediment exposure. Additionally, the effect of this chemical on acute stress responses, tolerance to hypoxia, and upper lethal temperature tolerance for fish acclimated to 15°C was measured (McLeay et al. 1983) to determine the extent to which Arctic grayling may respond to this reference toxicant under the ascribed conditions.

Prolonged Exposure Studies

Eight acrylic test streams (210 × 13 × 20 cm) placed side by side were used to expose *T. arcticus* to duplicate concentrations of 0, 100, 300, and 1000 mg suspended inorganic sediment$\cdot L^{-1}$. Removable screens were located at 100 and 200 cm from the head of each stream. A vertical overflow standpipe (for the continuous replacement of each test suspension) and horizontal outflow pipe (for the recirculation of each test suspension) were located at the downstream end and a horizontal inflow pipe at the upstream end. Each stream was covered by a fibreglass screen to prevent fish escaping. Water (±sediment) was pumped (Cole-Parmer impeller driven pumps) at $10\ L \cdot min^{-1}$ through 2-cm I.D. plastic tubing, thus maintaining a riffle area within the first third of each stream. Oil-free compressed air was continuously introduced at the head and midstream positions of each stream. Four 220-L polyethylene barrels were used as reservoirs. One barrel contained clear freshwater (control), while the sediment strengths (prepared daily) were stirred continuously in each of the other three barrels (Greey-Lightnin motor, 1700 rpm, stainless steel shaft and impeller).

Throughout the 6-wk exposure period the contents of each barrel were pumped continuously (Cole-Parmer Masterflex peristaltic pump) to each of two streams at $70\ mL \cdot min^{-1} \cdot stream^{-1}$, providing a 95% molecular exchange of each test suspension (and control water) within each stream every 24 h (Sprague 1973). Water temperature, dissolved oxygen, pH, conductance, NFR, and turbidity (formazin turbidity units; FTU) were monitored daily in each test stream. Nonfilterable residue values were determined for water samples taken from upstream, midstream, or downstream positions in each test stream. These values did not differ appreciably for any of the sediment strengths examined. Mean NFR values ($N = 50$) for the duplicate streams with a nominal strength of $100\ mg \cdot L^{-1}$ were 86 ± 25 and 93 ± 26 $mg \cdot L^{-1}$; for 300 $mg \cdot L^{-1}$, they were 286 ± 48 and 273 ± 51 $mg \cdot L^{-1}$; and for 1000 $mg \cdot L^{-1}$, they were 988 ± 130 and 955 ± 127 $mg \cdot L^{-1}$. Nonfilterable residue values for each of the control streams were consistently below the level of detection ($<5\ mg \cdot L^{-1}$). Temperatures for individual streams varied from 14.0 ± 0.4 to 15.8 ± 0.4°C ($N = 50$); dissolved oxygen from 9.3 ± 0.2 to 9.7 ± 0.3 $mg \cdot L^{-1}$; pH from 6.5 ± 0.2 to 6.7 ± 0.2; conductance from 25 ± 4 (control stream) to 36 ± 5 $\mu S \cdot cm^{-1}$ ($1000\ mg \cdot L\ stream^{-1}$); and turbidity from 1.1 ± 0.4 (control stream) to 1661 ± 224 FTU ($1000\ mg \cdot L\ stream^{-1}$). The water inflow rates ($N = 48$) to each stream remained relatively constant and ranged from 69 ± 3 to 73 ± 3 $mL \cdot min^{-1}$.

All apparatus was housed in a temperature-controlled room. Four 40-W broad-spectrum "Vitalite" fluorescent tubes illuminated each test stream for an 18-h light (1076 ± 54 lx) : 6-h dark sequence (synchronized incandescent 40-W bulbs, controlled by rheostat, provided a 0.5-h dawn and dusk period).

Fish feeding, growth, and behaviour

After a 3-wk acclimation to laboratory conditions of water and feed, 480 individually weighed juvenile *T. arcticus* were placed at random into the test streams ($60 \cdot stream^{-1}$). Initially each stream received (for a 7-d period) sediment-free freshwater (dechlorinated Vancouver City tap water at 15 ± 1°C, <5 mg $NFR \cdot L^{-1}$) at a flow rate of $\geq 230\ mL \cdot min^{-1}$ (95% exchange in ≤ 9 h).

During the 7-d acclimation period to the test streams (and for 2 wk thereafter), the fish were fed Biodiet No. 1 (7% wet body weight $\cdot d^{-1}$). This ratio was subsequently changed to a mixture of Biodiet No. 1 and No. 2 (1 : 1) and, after 6 wk, to Biodiet

No. 2. Equal quantities of live *Daphnia* sp. were also fed daily to fish in each stream. All food offered was spread evenly along the stream.

All fish were reweighed after the 7-d acclimation period. The predetermined concentration of sediment suspensions and freshwater (control) were then metered continuously from the reservoirs which were maintained on a daily basis to provide suspended sediment strengths within each test stream of 0 (control), 100, 300, and 1000 mg NFR$\cdot L^{-1}$.

The growth of fish in each test stream was monitored weekly throughout the subsequent 6-wk test period by weighing each test fish in a preweighed beaker containing a sample of streamwater to which the fish was exposed.

Information on the horizontal distribution of the test fish within each stream was obtained by inserting screens across each stream and determining the number of fish occupying four equal stream sections. The number of fish occupying each section was counted and the partitions removed. This procedure was performed after 4 and 5 wk of exposure to sediment.

The effect of suspended sediment on feeding response times of the grayling offered live food was determined after 5 or 6 wk of exposure. For feeding trials with surface drift, three fish were selected at random and confined within the downstream half of the stream by a screen. The fish were slowly moved to the most downstream portion of this half of the stream after a 60-min adjustment period. A live adult fruitfly (*Drosophila melanogaster*) was placed on the water surface 60 cm upstream and the fish permitted to move freely in the stream. The time for fish to consume this surface-drifting prey was recorded. All tests were terminated after 360 s. Three separate trials with surface drift were conducted, using different groups of fish. Five to nine replicates were carried out for each trial. Five to seven replicates per trial using naive grayling (stock, unexposed) were also carried out in the same manner, after their exposure to each test suspension in the downstream half of each stream for 1 h only.

The feeding response to live subsurface drift (*Artemia salina*) or benthic invertebrates (tubificid worms) employed the use of submersible nylon mesh baskets (60 × 15 × 15 cm). Unlike surface drift trials, in the response trials with benthic invertebrates all fish were restrained in the upper half of the test stream while 10 tubificids were distributed randomly along the basket which occupied the downstream half. Three grayling were then placed in the downstream section and the basket raised to permit very rapid observations of numbers of unconsumed tubificids after 1, 3, 5, 10, 15, 30, 45, and 60 min. Three trials per stream were carried out with different groups of fish, together with one trial using naive (unexposed stock) fish. Three feeding response trials with subsurface drift (live brine shrimp, *A. salina*) were also carried out after 6 wk of sediment exposure of the test fish. As in the feeding trial with benthic prey, 10 food organisms were used and the same procedure followed; however, the feeding response to subsurface drift for naive grayling was not examined.

Fish condition and tolerance to challenges

Following 6 wk of sediment exposure, 10 grayling were removed from each test stream to assess their condition. Length and weight (wet, dry) were determined and blood samples taken for determination of hematocrit, leucocrit, and glucose content. The condition factor and percentage body moisture content of each fish were calculated.

The effect of prolonged exposure to suspended sediment concentrations $\leq$1000 mg$\cdot L^{-1}$ on fish performance was assessed by a number of techniques. The acute lethal tolerance to the reference toxicant pentachlorophenol was established for groups of 10 fish from each stream, removed after 3 and 6 wk of exposure (McLeay et al. 1984). In addition, the effects of previous (6 wk) sediment exposure upon fish respiration (acute tolerance to hypoxia) was assessed using the sealed jar bioassay technique (McLeay 1976; Gordon and McLeay 1977). Finally, the thermal tolerance of groups of fish held in each test stream for 6 wk was compared using critical thermal maxima bioassays (McLeay and Howard 1977; McLeay and Gordon 1980).

Statistical Analysis

Condition factor (K) was determined according to Carlander (1969) where $K = cW\cdot L^{-3}$, assuming c is a constant (100), W = weight (grams), and L = fork length (centimeters).

The acute median lethal concentrations (96-h LC50 values) for grayling exposed to inorganic or organic sediment or to the reference toxicant were calculated (with 95% confidence intervals) using the computerized program of Stephan (1977). Times to death of 50% of grayling (LT50 values) exposed to pentachlorophenol following their prolonged exposure to sediment were calculated by log-probit analysis (Litchfield 1949). The median effective concentration (EC50 value) of sediment suspensions causing a net significant response for 50% of the exposed fish was calculated for the sublethal bioassays according to established procedures (Sprague 1968; McLeay and Howard 1977; McLeay and Gordon 1980).

Analysis of variance (ANOVA) followed by Dunnett's test (Zar 1974) was used to examine the fish growth data for significant differences.

Results and Discussion

Sediment and Water Quality

The measured concentrations of sediment to which fish were exposed remained relatively constant (and within the range of nominal strengths) throughout the acute and chronic exposure periods (McLeay et al. 1983, 1984). Accordingly, we consider that it is appropriate to assess the bioassay results in relation to nominal sediment concentrations. Those short-term variations in suspended sediment strengths which did occur were minor when compared with the suspended sediment variations that occur in streams subjected to placer mining (McLeay et al. 1983; Simmons 1984; Wagener 1984; Lloyd 1985). Furthermore, effects on aquatic organisms exposed to sediment generated from placer mining may be due to numerous factors besides sediment concentration (e.g. particle size, shape, hardness) (McLeay et al. 1983). Comparison of our particle size data for the inorganic sediments used, with that for natural or placer mine generated sediment suspensions in Alaskan streams (Lloyd 1985), indicates that our test sediments were similar in this respect.

With the exception of suspended sediment loadings, the quality of water used in these laboratory studies was compatible with fish survival. In the higher concentrations of sediment examined there was a slight but consistent elevation in conductivity. However, this would not cause any osmotic stress to test fish. Other water quality variables including temperature, pH,

TABLE 1. Summary of the laboratory findings for underyearling Arctic grayling acutely exposed to suspensions of placer mining sediment.

Test variable	Acclimation temperature (°C)	Sediment Type	Sediment Exposure (h)	Response elicited	EC50[a] ($g \cdot L^{-1}$)
Survival	15	Inorganic	384	No effect	>50
	15	Inorganic	96	No effect	>250
	15	Organic	96	No effect	>50
	5	Inorganic	96	10 and 20% mortality in 20 and 100 $g \cdot L^{-1}$, respectively	>100
Gill histology	15	Inorganic	96	No effect	>100
	15	Organic	96	No effect	>50
Hematocrit	15	Inorganic	24, 96	No effect	>100
	15	Organic	24	No effect	>20
	5	Inorganic	96	No effect	>100
Leucocrit (number of white blood cells)	15	Inorganic	24	Decrease	52
	15	Organic	24	Decrease	5.8
	5	Inorganic	96	No effect	>100
Blood sugar (plasma glucose)	15	Inorganic	24	Increase	—[b]
	15	Organic	24	Increase	<0.05
	5	Inorganic	96	Increase or more variation	—[b]
Tolerance to hypoxia (time to death)	15	Inorganic	5	Increase	4.4
	15	Organic	5	Decrease	0.2
	5	Inorganic	8	No effect	>100
Temperature Tolerance (critical thermal maxima)	15	Inorganic	12	Slight decline	0.1
	15	Organic	12	Slight decline	8.5
	5	Inorganic	20	No effect	>100
Distribution/ behaviour	15	Inorganic	96	Swimming at surface	<10
	5	Inorganic	96	Swimming at surface	<10
	15	Organic	96	No effect	>50
	5	Organic	96	No effect	>50

[a]Median effective concentration causing a net significant response for 50% of fish.
[b]Unable to determine due to increased variance of data.

and dissolved oxygen were compatible with fish survival and, in the case of the prolonged exposure studies, optimal or near optimal for their growth and long-term well being.

Fish Survival

All 15°C-acclimated grayling survived a 96-h exposure to suspensions of inorganic sediment up to and including 250 $g \cdot L^{-1}$, and also survived a 16-d exposure to 50 $g \cdot L^{-1}$. Similarly, all grayling acclimated to 15°C survived a 96-h exposure to organic sediment up to and including 50 $g \cdot L^{-1}$ (Table 1). In experiments carried out at 5°C, 10 and 20% mortality was recorded for inorganic sediment strengths of 20 and 100 $g \cdot L^{-1}$, respectively. All 5°C-acclimated fish survived a 96-h exposure to lower concentrations of suspended sediment.

All 60 fish in each test stream survived the initial 7-d acclimation period. Some mortalities, unrelated to sediment strength, occurred thereafter in each stream. Overall, 8% of the test fish died during the 6-wk exposure (control, 8 and 13%; 100 $mg \cdot L^{-1}$, 5 and 8%; 300 $mg \cdot L^{-1}$, 5 and 10%; 1000 $mg \cdot L^{-1}$, 5 and 12% (Table 2). Gills of the dead fish showed no signs of clubbing, accumulation of sediment, or excessive mucous production. Most of these fish showed signs of physical injury (nips, tears), presumably related to aggressive interactions.

Our studies indicate that underyearling *T. arcticus* can survive long-term (6 wk) exposure to inorganic sediment concentrations ≤1000 $mg \cdot L^{-1}$ and short-term (4 d) exposure to high inorganic (≤250 $g \cdot L^{-1}$) or organic (≤50 $g \cdot L^{-1}$) sediment concentrations under otherwise optimal water quality conditions. The <20% mortalities in grayling acclimated to 5°C and held for 4 d in 20 or 100 $g \cdot L^{-1}$ suspensions of inorganic sediment suggest a decreased tolerance for fish acclimated to cold water. However, confirmation of this would require further study. In the chronic exposure experiments, fish mortalities were considered to be a result of aggression between individuals rather than a response to suspended sediment concentrations. We are not aware of any comparable data relating to the lethal tolerance of

TABLE 2. Summary of the laboratory findings for underyearling Arctic grayling exposed for up to 6 wk (1008 h) to suspensions of placer mining sediment.

Test variable	Exposure period (h)	Sediment concentration ($mg \cdot L^{-1}$)	Observation
Survival	1008	0	87 and 92% survival
		100	92 and 95% survival
		300	90 and 95% survival
		1000	88 and 95% survival
Physiological condition	1008	0–1000	No effect on hematocrit, leucocrit, or plasma glucose
			No effect on condition factor or body moisture
			No overt signs of disease or gill damage
		300	Fish slightly paler than controls
		1000	Fish notably paler than controls; indistinct parr marks
Growth (weight gain)	1008	0	241% increase
		100	227% increase; 6% reduction relative to controls
		300	217% increase; 10% reduction relative to controls
		1000	161% increase; 33% reduction relative to controls
Tolerance to hypoxia	1008	0–1000	No effect on residual dissolved oxygen content at death
		300, 1000	Significant reduction in times to death (elevated oxygen consumption rates) relative to fish in 0 and 100 mg sediment$\cdot L^{-1}$
Temperature tolerance	1008	0–1000	No effect on critical thermal maxima
Tolerance to pentachlorophenol	1008	100	No effect on acute lethal tolerance
		300, 1000	Decreased acute lethal tolerance
Feeding response to surface drift	840	0	Rapid feeding; 100% consumption in 6–8 s
		100	Decreased mean response times (16–32 s) relative to controls
		300	Decreased times (24–82 s) relative to lower sediment concentrations; miss-strikes common
		1000	Decreased times (25 to >191 s) relative to lower sediment concentrations; miss-strikes common
Feeding response (naive fish) to surface drift (*D. melanogaster*)	1	0	100% consumption in 10 ± 5 to 25 ± 29 s
		100	Decreased response times (10 to >360 s); failure to feed in 3 of 24 tests
		300	Decreased response times (14 to >360 s); failure to feed in 6 of 24 tests; some miss-strikes
		1000	Decreased response times (209 to >360 s); failure to feed in 18 of 24 tests; some miss-strikes
Feeding response to subsurface drift (*A. salina*)	1008	0	Mean times to consume all prey 3–4 min
		100	Mean time to consume all prey 3 min
		300	Mean times to consume all prey 4–8 min
		1000	Failure to consume all prey items in 60 min
Feeding response to benthic invertebrates (tubificids)	840	0	Mean time to consume all prey 9 min
		100	Mean times to consume all prey 4–9 min
		300	Mean times to consume all prey 6–9 min
		1000	Failure to consume all prey items in 60 min

TABLE 2. (*Concluded*)

Test variable	Exposure period (h)	Sediment concentration ($mg \cdot L^{-1}$)	Observation
Feeding response (naive fish) to benthic invertebrates (tubificids)	1	0	Mean time to consume all prey 5 min
		100	Mean times to consume all prey 5–10 min
		300	Mean times to consume all prey 3–5 min
		1000	Failure to consume all prey in 60 min
Distribution in experimental streams	672 – 840	0	60% of fish in upstream section
		100	32% in upstream section; downstream displacement
		300	15% in upstream section; downstream displacement
		1000	16% in upstream section; downstream displacement

T. arcticus to suspended sediment under controlled conditions. Recently, the survival and health of *T. arcticus* held within cages in streams being actively placer mined have been reported (McLeay et al. 1983; Simmons 1984). These field studies, carried out at two locations (Yukon and Alaska), confirm that juvenile Arctic grayling can withstand short-term (9 d) exposure to high (6.6 $g \cdot L^{-1}$) total sediment residue.

Available information concerning the lethal tolerance of other salmonid species to sediment is sparse and somewhat inconsistent. Smith (1978) reported that high concentrations (28–55 $g \cdot L^{-1}$) of suspensions prepared from two natural sediment sources were required to kill chum salmon (*Oncorhynchus keta*) fry within 4 d. On the other hand, Herbert and Merkens (1961) reported that a 10- to 15-d exposure of juvenile rainbow trout (*Salmo gairdneri*) to suspensions of kaolin or diatomaceous earth as low as 270 $mg \cdot L^{-1}$ caused significant mortalities of test fish. Noggle (1978) determined that the acute lethal tolerance (96-h LC50 values) to suspensions of natural sediments for groups of wild or hatchery-reared juvenile coho salmon (*O. kisutch*), chinook salmon (*O. tshawytscha*), or steelhead trout (*S. gairdneri*) varied from 1.2 to 35 $g \cdot L^{-1}$. Differences noted were attributed largely to seasonal temperature variations, with a lower tolerance of fish to sediment observed during the summer months. This conclusion was supported by a report of lower lethal concentrations of natural sediments for nonsalmonid fishes, with higher test temperatures (Rogers 1969). More recently, differences in the acute lethal tolerance of juvenile sockeye salmon exposed to various-sized fractions of Fraser River sediment were found (J. A. Servizi, International Pacific Salmon Fisheries Commission, Sweltzer Creek Laboratory, Cultus Lake, B.C., pers. comm.). The variability in lethal tolerance of fish to suspended sediment, as apparent from these and the present studies, may be partially explained by differences in the nature (e.g. particle size, angularity, hardness) of the suspended sediment material used in the tests (Anonymous 1965).

Physiological Condition

After the acute (4 d) exposures, fish showed no gross evidence of lesions or signs of disease. Gills appeared normal, and no lamellar clubbing, hypertrophy, or hyperplasia was found associated with short-term exposure to inorganic (<100 $g \cdot L^{-1}$) or organic (≤50 $g \cdot L^{-1}$) sediment. Similarly, fish held for up to 6 wk in test streams containing suspended sediment strengths ≤1000 $mg \cdot L^{-1}$ appeared healthy. Fins and opercula of these fish were normal and no internal or external hemorrhages or lesions were observed. Gross examination of the gills revealed no clubbing, discolouration, excess mucus productions, or adhesion of sediment particles. However, all fish in 1000 mg suspended sediment$\cdot L^{-1}$ had indistinct parr marks and were notably paler than control fish. Fish held in 300 mg sediment$\cdot L^{-1}$ were slightly paler than control fish (Table 2).

As in our studies, Smith (1978) found no damage to gills of juvenile chum salmon acutely exposed to high (up to 55 $g \cdot L^{-1}$) concentrations of suspended inorganic sediment. In contrast, a number of investigators have reported histopathological changes in fish gills attributable to sediment exposure. Herbert and Merkens (1961) observed thickening and fusion of secondary gill lamellae of some rainbow trout exposed for several weeks to diatomaceous earth or china clay. Noggle (1978) reported notable gill histopathologies in certain juvenile salmonid fish held in inorganic sediment suspensions ≤13 $g \cdot L^{-1}$ for up to 96 h. Noggle (1978) also reported thickening and fusion of gill lamellae in trout held in suspensions of diatomaceous earth for up to 96 h. Gill damage was noted in juvenile sockeye salmon exposed for 96 h to 3 g fine sediment$\cdot L^{-1}$ from the Fraser River (J. A. Servizi, pers. comm.). Under field conditions, Simmons (1984) recorded histopathological changes in the gills of Arctic grayling exposed for 48–96 h to high concentrations of suspended placer mining sediment. Birtwell et al. (1984) also documented the presence of histopathological changes in the gills of *T. arcticus* of different age classes within a drainage basin being placer minded. But, unlike Simmons' (1984) findings, these (Birtwell et al. 1984) results with wild uncaged fish could not be conclusively attributable to exposure to placer-mining sediment. Pickral (1981) cited the variability in reported findings of fish gill tissue damage associated with high concentrations of suspended sediment and suggested that the evidence for such an effect (attributable to sediment concentration) was inconclusive. We interpret this variability in response to differing sediment characteristics (particle shape, size, hardness), perhaps in conjunction with biological (fish age, size, prior history of exposure) and experimental (exposure period, test apparatus) differences. Our observation that fish exposed to 300 and 1000 $mg \cdot L^{-1}$ were paler in colour is consistent with other findings for salmonid fish exposed to elevated suspended sediment levels under field or laboratory conditions (Herbert and Merkens 1961; Herbert et al. 1961) and probably reflects a contraction of epithelial chromatophores in response to background colour.

Blood hematocrit values for all groups of fish acclimated to 15 or 5°C and exposed acutely (1−4 d) to differing concentrations of inorganic or organic sediment were unchanged from those for the respective groups of control fish (Table 1). Mean values for fish acclimated and tested at 15°C were 31−35% whereas those for grayling acclimated and tested in colder (5°C) water were somewhat lower (28−30%). Sample variances were small. Hematocrit values for groups of grayling held for 6 wk in suspended sediment concentrations ≤1000 $mg \cdot L^{-1}$ were also unchanged from those for corresponding control groups.

As in our studies, Noggle (1978) reported that hematocrit values of underyearling coho salmon were unchanged by holding fish for 96 h in suspended sediment concentrations equivalent to 0.8 of the 96-h LC50 value. Other studies with non-salmonid fishes given short-term exposure to sediment have shown unchanged, elevated, or depressed hematocrit values (Berry 1973; Neumann et al. 1975).

Hypoxic conditions cause significant increases in hematocrit values for salmonids (Holeton and Randall 1967; Swift and Lloyd 1974; Soivio et al. 1974a, 1974b; Casillas and Smith 1977). On the other hand, changes in hematocrit are somewhat resistant to acute stress, including that caused by exposure of fish to sublethal concentrations of a variety of aquatic contaminants (McLeay and Gordon 1977, 1979, 1980). The absence of any changes in hematocrit for grayling exposed to suspended sediment for brief periods (1−4 d) or for periods as long as 6 wk suggests that these exposures did not lower blood oxygen tension. O'Connor et al. (1977) reported elevated hematocrit levels, red blood cells counts, and hemoglobin values together with histological evidence of gill damage for certain species of estuarine fish exposed to suspended sediment whereas for other species these blood values and gill histology were unchanged from controls. A lack of change in hematocrit values does not necessarily imply no gill tissue damage, however, since fish have a large "reserve" surface area of gill tissue available for maintaining blood gas tensions at normal values (Randall 1970). The decrease in hematocrit values noted in this study for Arctic grayling acclimated to 5°C, relative to those acclimated to warmer (15°C) water, is consistent with reported findings for other salmonid species (Banks et al. 1971).

In the acute stress tests with sediment, mean blood leucocrit values for all groups of 15°C-acclimated fish exposed for 24 h to inorganic or organic sediment concentrations ≥1000 $mg \cdot L^{-1}$ were decreased relative to values for corresponding control groups. Median effective concentrations causing a net significant response for 50% of test fish varied from 5.8 (organic sediment) to 52 $g \cdot L^{-1}$ (inorganic sediment) (Table 1). Leucocrit values for the coldwater-adapted fish were somewhat lower and were not altered by sediment exposure.

Blood sugar values for groups of fish acclimated to both 5 and 15°C were affected by acute (1−4 d) sediment exposure. Values for sediment-exposed fish were generally increased and/or more variable, relative to corresponding control values. Suspensions of organic sediment consistently elevated blood sugar levels at all concentrations examined (≥50 $mg \cdot L^{-1}$). Changes due to inorganic sediment concentrations ≥500 $mg \cdot L^{-1}$ were also evident at both temperatures tested, although increased variances due to treatment prevented the calculation of EC50 values (Table 1).

Leucocrit values for groups of fish held for 6 wk in suspended sediment strengths ≤1000 $mg \cdot L^{-1}$ were unaffected by treatment (range 0.9 ± 0.2 to 1.0 ± 0.4%). Mean plasma glucose values for these fish were consistently low (58−68 mg%), with no consistent changes in magnitude or sample variance attributable to sediment exposure. The percentage moisture content for these fish groups (means 77−79%) was also unaffected by sediment exposure.

Unlike hematocrit, leucocrit values of salmonids can change rapidly and dramatically in response to stress (McLeay 1975; McLeay and Gordon 1977;Wedemeyer and McLeay 1981; Wedemeyer et al. 1984). Short-term exposure of rainbow trout or coho salmon to concentrations of aquatic contaminants as low as 0.1 of the 96-h LC50 can cause significant declines in leucocrit, provided that test fish are in good condition and unstressed beforehand (McLeay and Howard 1977; McLeay and Gordon 1979, 1980). The general decline in leucocrit values for warmwater-acclimated grayling held in suspensions of inorganic and organic sediment for 24 h indicates that each of these sediment types was stressful to these fish. Values for control groups were similar to those found previously for underyearling coho salmon, and somewhat elevated from control values for rainbow trout (McLeay and Gordon 1977). The absence of a consistent leucocrit response for the coldwater-acclimated grayling may reflect the influence of prior stress (i.e. disturbances to all control and test fish during the 4-d exposure period) or perhaps a differing mechanism of response to stress for coldwater- versus warmwater-acclimated fish.

The stress reactions (depressed leucocrit values, elevated plasma glucose values) found in our studies with grayling exposed for 24 h to sublethal strengths of the reference toxicant pentachlorophenol are typical of the acute responses of underyearling rainbow trout to this chemical (McLeay and Gordon 1980). Diverse environmental stressors including sublethal strengths of aquatic contaminants are known to cause a rapid elevation and/or increased variance in blood sugar levels of salmonids (McLeay 1977; Wedemeyer and McLeay 1981). A consideration of the blood sugar changes found in our tests, together with leucocrit changes for sediment-exposed fish, confirms that sublethal strengths of the inorganic and organic sediment suspensions examined were indeed acutely stressful to Arctic grayling. The variation in plasma glucose levels noted for coldwater-acclimated grayling exposed to suspensions of inorganic sediment for 96 h indicates that grayling are stressed by this sediment, regardless of season or acclimation temperature.

Other studies of the hematological effects of sediment suspensions on fish are limited. Noggle (1978) found that blood sugar values for groups of coho salmon held for 96 h in inorganic sediment suspensions ≥0.2 LC50 were significantly changed from those for control fish. O'Connor et al. (1977) reported that the laboratory exposure of a number of species of estuarine fish to suspensions of natural sediments caused hematological changes indicative of stress responses. Similarly, J. A. Servizi (pers. comm.) recorded elevated blood sugar levels in adult sockeye salmon exposed for 96 h to natural sediments (≥500 $mg \cdot L^{-1}$).

Exposure of Arctic grayling and other fish species to suspended sediment at strengths which can be found within natural streams has been shown to cause a number of typical stress responses, including short-term increases in plasma cortisol levels (Redding and Schreck 1980), depletion of liver glycogen energy reserves (Sherk et al. 1974; O'Connor et al. 1977), and elevation of plasma glucose levels (Noggle 1978). When the stress is prolonged, certain fish stress indices including plasma cortisol and glucose levels and numbers of circulating white blood cells (leucocrit) recover to basal or near-basal levels (McLeay and Brown 1974; McLeay 1977; McLeay and Gordon 1977; Redding and Schreck 1980). These recoveries typify the stage of resistance to stress (Selye 1950), and are only achieved at a metabolic cost. Thus the apparent absence of change in leucocrit or plasma glucose values for Arctic grayling following 6 wk of exposure to suspensions of placer mining sediment is not unexpected, and does not imply that these fish were not stressed. A better understanding as to whether the grayling in this study were chronically stressed by suspended sediment, and the stage of this response after 6 wk of exposure (i.e. resistance or exhaustion stages; Selye 1950), would require a more detailed biochemical/histopathological examination of test fish.

Condition Factors and Growth

Fish condition factors after 3 and 6 wk of sediment exposure did not differ significantly due to treatment. Mean (±SD) condition factors after 6 wk varied from 0.81 ± 0.11 to 0.92 ± 0.11. These values are similar to those for underyearling Arctic grayling captured from clear water Yukon streams (Birtwell et al. 1984). As in the present study, Sigler (1981) found no change in condition factors for underyearling steelhead trout or coho salmon held for extended periods in laboratory streams containing sediment suspensions whereas these fish grew less than the control fish in clear water.

The initial mean weight for each group of 60 grayling in each stream was similar (range 0.52 ± 0.11 to 0.56 ± 0.1 g). However, for each of the subsequent exposure periods, the mean weight for each group of control fish was greater than that for any sediment treatment, and reduced growth due to sediment exposure was concentration dependent. The average percentage weight gains, based upon the mean weight of fish at 0 versus 6 wk of exposure, were as follows: 0 $mg \cdot L^{-1}$, 241%; 100 $mg \cdot L^{-1}$, 227%; 300 $mg \cdot L^{-1}$, 217%; 1000 $mg \cdot L^{-1}$, 161%. Relative to the control values, there was a 33, 10, and 6% reduction in growth of fish exposed to a suspended sediment concentration of 1000, 300, and 100 $mg \cdot L^{-1}$, respectively, during the test period (Table 2).

According to ANOVA, fish weights differed significantly ($P < 0.05$) due to treatment for all exposures of 2 wk or greater. Dunnett's test showed that weights for fish exposed to 1000 mg sediment$\cdot L^{-1}$ were less than those for corresponding groups of control fish at 2, 3, 4, 5, and 6 wk. Weights for fish held in 100 and 300 mg suspended sediment$\cdot L^{-1}$ were significantly less than corresponding controls at 3, 4, and 5 wk only. Differences between values for the lower sediment strengths and controls did not differ significantly at 6 wk, presumably due to increased sample variances.

The significant reduction in growth of grayling exposed to suspended placer mining sediment at 100, 300, and 1000 $mg \cdot L^{-1}$ is consistent with previous findings. Herbert and Richards (1963) reported growth impairment for rainbow trout reared for 33–40 wk in suspensions of coal-washery waste or wood fibre as low as 50 $mg \cdot L^{-1}$. Similarly, Sigler et al. (1984) found reduced growth for underyearling steelhead trout or coho salmon held for 2–3 wk in laboratory streams containing suspended clay solids with turbidity values as low as 25 FTU. Depending on the nature and availability of the food supply, settled sediment fines may also restrict the growth of stream-reared fish (Crouse et al. 1981).

Unlike grayling in test streams receiving 1000 mg suspended sediment$\cdot L^{-1}$, those reared in 100 or 300 mg sediment$\cdot L^{-1}$ grew nearly as well as the control fish. However, all fish were presented with an abundant supply of food throughout the test period. These lower suspended sediment strengths, if present for extended periods within natural streams where food supply was limited, could result in a greater impairment of fish growth than we observed.

The reduced growth of sediment-exposed grayling found in our study likely reflects a reduced scope for growth. A sustained increase in swimming activity of fish due to suspended sediment (Berg 1982), or other increased metabolic demands, would increase energy costs and result in less energy available for growth. Decreased growth may also be due to a decreased food intake, although it is thought that this would be reflected in a reduction in fish condition factor. As in the present instance, other studies with salmonids exposed to sublethal concentrations of aquatic contaminants have reported reduced growth, unaccompanied by any change in fish condition factors or percentage moisture content (although food conversion efficiency was reduced) (Webb and Brett 1972, 1973). A reduction in food conversion efficiency, if caused by increased maintenance energy costs, would reduce the proportion of energy available for growth (Warren and Davis 1967), i.e. reduce the "scope for growth" (Brett 1976).

Tolerance to Hypoxic Challenge

Mean times to death for groups of 15°C-acclimated fish held briefly (hours) in sealed jars containing suspended sediment increased progressively with increasing inorganic sediment concentation; however, mean residual oxygen values for each treatment did not differ from those of controls. At 5°C, inorganic sediment did not affect either times to death or residual oxygen values at death. Contrasting with this situation, the times to death for groups of grayling held in suspensions of organic sediment decreased progressively with increasing sediment strength (Table 1). Residual oxygen values for these groups were variable and again showed no consistent increase with sediment concentration.

Sealed jar bioassays performed after 6 wk of exposure of grayling to suspended sediment concentrations ≤1000 $mg \cdot L^{-1}$ indicated that their short-term capacity to withstand hypoxic conditions was unimpaired. Mean residual dissolved oxygen values derived for fish from streams receiving identical treatment were somewhat variable and showed no consistent trend with respect to sediment strength. However, unlike these findings, the times to death of fish exposed to 300 and 1000 mg sediment$\cdot L^{-1}$ were decreased consistently and significantly ($P < 0.05$) relative to control fish or those held in 100 mg sediment$\cdot L^{-1}$. The LT50 values were as follows: 0 $mg \cdot L^{-1}$, 147 and 158 min; 100 $mg \cdot L^{-1}$, 145 min; 300 $mg \cdot L^{-1}$, 115 and 125 min; 1000 $mg \cdot L^{-1}$, 125 min. Oxygen uptake rates calculated for each group of 10 fish were between 2.1 and 2.2 mg $O_2 \cdot g$ $fish^{-1} \cdot h^{-1}$ for control fish and those exposed to 100 mg sediment$\cdot L^{-1}$ whereas those for fish exposed to 300 and 1000 mg suspended sediment$\cdot L^{-1}$ were 2.3–2.5 mg $O_2 \cdot g$ $fish^{-1} \cdot h^{-1}$. These data indicate an increase in oxygen consumption rates for each group of fish exposed to the two higher strengths of suspended sediment only.

The tolerance of juvenile *T. arcticus* to hypoxia was, in general, similar to that found previously for underyearling coho salmon or rainbow trout examined under identical test conditions (McLeay 1976; Gordon and McLeay 1977). The critical residual dissolved oxygen level at which each of these fish species dies, if acclimated at 15°C and held in freshwater at 20°C, is approximately 2.0 mg $O_2 \cdot L^{-1}$. The even greater tolerance to hypoxic conditions found in the bioassays with grayling acclimated to 5°C and tested at 10°C (critical value approximately 1.5 mg $O_2 \cdot L^{-1}$) is also consistent with findings for other salmonid fish in response to a decrease in test temperature (Gordon and McLeay 1977) and for *T. arcticus* under field testing conditions (Simmons 1984). Simmons carried out identical sealed jar bioassay experiments using grayling that had been exposed to clear (unmined) and turbid (mined) creekwater in Alaska for 48 or 96 h. He found no significant differences in mean times to death (equilibrium loss) between streams but a significantly lower residual dissolved oxygen concentration between test conditions at 8–9°C and those at 3–4°C.

The progressive decline in times to death found for groups of grayling exposed to higher concentrations of organic sediment is likely due to the high oxygen demand demonstrated for this organic sediment (McLeay et al. 1983). Unlike this response, the increase in mean times to death for 15°C-acclimated grayling, with increasing strengths of inorganic sediment, suggests a reduction in respiratory rate attributable to this sediment. This response could be due to increased swimming activity of fish in the clear solution, in response to visual "disturbance" during the bioassay. It could also reflect decreased physical activity, reduced ventilatory rate, or decreased efficiency of oxygen transfer, caused by progressively higher strengths of inorganic sediment. The significance of this response is unclear in view of the lack of effect of these sediment suspensions on the fishes' tolerance to hypoxia and on the absence of a time-to-death response to inorganic sediment for grayling acclimated to cold water.

The effect of suspended sediment on the respiration rate of fish has not been examined to any extent. Neumann et al. (1975) reported no change in the respiratory rates of oyster toadfish (*Opsanus tau*) held briefly in a 2 $g \cdot L^{-1}$ suspension of natural sediment, although a 72-h exposure to 11 g sediment$\cdot L^{-1}$ caused a great variance in oxygen uptake rates compared with control fish.

Aquatic contaminants known to affect fish respiration cause a concentration-dependent increase in residual oxygen levels of salmonid fishes held in sealed jars (McLeay 1976; Vigers and Maynard 1977) whereas those contaminants known to exert their toxic effects otherwise may not elicit this response (McLeay and Gordon 1980). The absence of significant changes in residual oxygen values for grayling held in suspensions of inorganic or organic sediments suggests either that these sediments do not impair their tolerance to hypoxic conditions or that the strengths of sediment to which fish were exposed were too low to evoke a response. The elevated residual oxygen values for grayling held in sublethal strengths of pentachlorophenol confirm that these fish will indeed show a response in this bioassay to a contaminant known to affect fish respiration.

The increased oxygen uptake rates found for grayling held for 6 wk in 100 or 3000 mg suspended sediment$\cdot L^{-1}$ may reflect an increased basal metabolic rate or a sustained increase in physical activity for these fish. This response indicates a decreased scope for activity of the fish (Fry 1971). Measurements of scope for activity have been used previously to assess the impact of environmental stressors on fish (Brett 1958; Wedemeyer and McLeay 1981). Since the ability of grayling to withstand hypoxia was unaffected by chronic exposure to any sediment strength examined (residual dissolved oxygen values were unchanged), the decreased times to death found in sealed jar tests for fish exposed to 300 or 100 mg sediment$\cdot L^{-1}$ probably reflect heightened energy demands for these fish rather than impaired blood gas exchange (D. J. Randall, Department of Zoology, University of British Columbia, Vancouver, B.C. pers. comm.). This conclusion is also consistent with the lack of change in hematocrit values for the sediment-exposed fish. Thus the present findings do not suggest a greater capacity to adapt to hypoxic conditions for this species than has been determined previously for other salmonid fish.

Tolerance to Elevated Temperature

Mean critical thermal maxima (upper lethal temperatures) for groups of fish acclimated to 15°C and subjected to progressive temperature increases in the presence of inorganic sediment were decreased slightly (<1°) but consistently from corresponding control values by concentrations of ≥500 $mg \cdot L^{-1}$ (EC50 100 $mg \cdot L^{-1}$; Table 1). Organic sediment concentrations ≥5 $g \cdot L^{-1}$ also caused a slight decline in mean critical thermal maxima (Table 1).

For fish groups acclimated to 5°C, upper lethal temperatures were unaffected by any inorganic sediment suspensions to which fish were exposed (Table 1). Mean critical thermal maxima for these fish groups (25°C) were decreased somewhat from those (27–28°C) for fish acclimated to 15°C.

Tolerance to elevated temperatures was not affected by a 6-wk exposure to any of the suspended sediment strengths examined (Table 2). Upper lethal temperatures for groups of fish from each stream receiving sediment were similar (range 27.7 ± 0.1 to 28.0 ± 0.2°C) and variances were low. These values were unchanged from that (27.7 ± 0.3°C) for fish from one control stream. Temperatures at death of fish from the second control stream were slightly lower (27.0 ± 0.7°C), a response consistent with their cooler acclimation temperature.

The temperature tolerance for Arctic grayling held in clear freshwater was similar to that determined for underyearling coho salmon or rainbow trout under identical conditions (McLeay and Howard 1977; McLeay and Gordon 1980). LaPerriere and Carlson (1973) reported earlier that the (high) thermal tolerance of various life stages of Arctic grayling was similar to other salmonid species. The increased resistance to high temperature with an increased temperature of acclimation (15 vs. 5°C) noted for grayling in our studies is also consistent with earlier findings for other salmonid species (Brett 1952; Black 1953). The seasonal photoperiod to which fish are acclimated can also influence thermal tolerance (McLeay and Gordon 1978). Thus, differences noted in temperature tolerance for grayling acclimated to 15 or 5°C probably reflect the effect of a number of variables (e.g. seasonal photoperiod, developmental stage of fish, fish condition) besides the temperature to which the fish were acclimated.

Sublethal concentrations of a number of aquatic contaminants (i.e. pulpmill effluent, herbicides, certain heavy metals) have been shown previously to cause a concentration-related decrease in the temperature tolerance of salmonid fish (McLeay and Gordon 1978, 1980). Some aquatic contaminants can lower the upper lethal temperature tolerance of salmonid fish by as much as 4–5° (McLeay and Gordon 1978, 1980). The minimal responses caused by exposing grayling to very high concentrations of suspended inorganic and organic sediment indicate that these sediment loadings do not interfere to a large extent with the immediate thermal adaptive capacity of grayling. These findings, considered together with findings of thermal tolerance effects noted previously for salmonid fish and other aquatic contaminants, suggest that short-term or prolonged (6 wk) exposure of juvenile grayling to high loadings of suspended sediment may not impair tissue respiration to a significant extent. Nevertheless, the threshold-effect (EC50) levels of 100 and 8471 mg organic sediment$\cdot L^{-1}$ determined for the 15°C-acclimated grayling indicate that a measurable reduction in critical thermal maxima for these fish was caused by these and higher sediment strengths.

Regardless, the findings indicate that the capacity of Arctic grayling to withstand high temperatures was virtually unaffected by short or prolonged exposure to any strength of suspended sediment examined. This result is consistent with previous evidence that little if any change in the upper lethal temperature tolerance of salmonid fish is caused by aquatic contaminants which do not block oxygen exchange at the gills or otherwise impair tissue respiration (McLeay and Gordon 1980; McLeay et al. 1983).

Tolerance to Reference Toxicant

The acute lethal tolerance to pentachlorophenol for groups of grayling acclimated to 5 and 15°C was similar; 96-h LC50 values were 61 and 67 $\mu g\cdot L^{-1}$, respectively. At both temperatures, all test fish survived exposures to pentachlorophenol concentrations ≤ 40 $\mu g\cdot L^{-1}$ with no overt signs of distress. Pentachlorophenol at sublethal strengths (0.4 and 0.7 of the 96-h LC50 value) reduced critical thermal maxima for grayling by 0.9 and 1.8° respectively. In sealed jar bioassays, a concentration-dependent increase in residual oxygen values, together with a progressive decrease in times to death of the test fish, occurred. Sublethal strengths of 35 $\mu g\cdot L^{-1}$ (0.5 of the 96-h LC50 value) and 50 $\mu g\cdot L^{-1}$ (0.7 LC50) elevated residual oxygen values (3.2 and 3.3 mg $O_2\cdot L^{-1}$, respectively) from those determined for each of the control groups (2.0 ± 0.3 and 2.0 ± 0.4 mg$\cdot L^{-1}$). Additionally, exposure of 15°C-acclimated fish for 24 h to pentachlorophenol concentrations of ≥ 20 $\mu g\cdot L^{-1}$ (0.3 of 96-h LC50) caused consistent increases in plasma glucose values and decreases in leucocrit values, with marked responses evident at higher (>35 $\mu g\cdot L^{-1}$) strengths. Hematocrit values for these fish were unchanged from control values.

After 6 wk of exposure to sediment, the tolerance to a 150 $\mu g\cdot L^{-1}$ solution of pentachlorophenol for the two control groups and both groups of fish reared in 100 mg sediment$\cdot L^{-1}$ was similarly high (LT50, 285–310 min). Unlike these values, times to death for fish exposed for 6 wk in 300 and 1000 mg sediment$\cdot L^{-1}$ were appreciably shorter and significantly different ($P < 0.05$) from the controls (Table 2). These results indicate that the acute lethal tolerance of grayling to this reference toxicant was depressed by prolonged (6 wk) exposure to these higher strengths of suspended sediment.

This decreased tolerance due to prolonged sediment exposure was not evident by 3 wk. Times to death in 150 μg pentachlorophenol$\cdot L^{-1}$ for groups of grayling reared in 0, 300, or 1000 mg sediment$\cdot L^{-1}$ for this shorter period did not differ significantly (LT50, 255–290 min). Times to death for fish exposed to this reference toxicant following 3 wk in 100 mg sediment$\cdot L^{-1}$ were slightly but significantly ($P < 0.05$) longer (LT50, 310 and 320 min) than those for control fish.

The lethal response of *T. arcticus* to the reference toxicant pentachlorophenol was similar to that recorded for other salmonid species. The 96-h LC50 values are within the range for this respiratory inhibitor reported previously for populations of healthy hatchery-reared rainbow trout or coho salmon fingerlings acclimated to, and tested in, 10–12°C water with pH and hardness characteristics similar to those used in the present studies (Davis and Hoos 1975; McLeay and Gordon 1980). Results from these bioassays, considered together with the observations of the fish stock, suggest that the condition and tolerance to aquatic contaminants of the laboratory-reared grayling were typical of healthy populations of young salmonid fish species.

The greater reduction in critical thermal maxima for grayling exposed to pentachlorophenol in this study was also consistent with that for other salmonid fishes challenged with this reference toxicant (McLeay and Gordon 1980). This finding indicates that the tolerance of grayling to temperature extremes is similarly influenced by this aquatic contaminant. Comparison of these data with the minimal or negligible response to suspended sediment confirms that the magnitude of effect for this response is dependent on both the nature and concentation of toxicant to which fish are subjected.

The respiratory responses of grayling to pentachlorophenol examined in relation to their tolerance of hypoxic conditions (elevated residual dissolved oxygen values, decreased times to death) are consistent with the response to this reference toxicant noted for underyearling rainbow trout tested under identical conditions (McLeay and Gordon 1980). This toxicant is known to increase the oxygen consumption rate of salmonid fish and is believed to uncouple mitochondrial respiration (Chapman and Shumway 1978).

When challenged with a lethal concentration of pentachlorophenol after 6 wk of exposure to 300 or 1000 mg suspended sediment$\cdot L^{-1}$, the fish were less able to withstand this reference toxicant than control fish or those held in 100 mg$\cdot L^{-1}$. This result is consistent with the results of the sealed jar bioassay tests in which fish in the higher sediment strengths died more rapidly than control fish or those in 100 mg sediment$\cdot L^{-1}$.

Fish Behaviour

Behavioural observations of fish during the acute exposure studies were in most instances prevented due to the opacity of the suspensions. For the more dilute concentrations of inorganic and organic sediment examined (50 and 100 mg$\cdot L^{-1}$), some signs of increased coughing and/or increased swimming activity of fish (relative to controls) were apparent; however, these were casual observations and no attempt was made to establish whether differences were significant. For the survival bioassays conducted at 5 and 15°C, all inorganic sediment concentrations ≥ 10 g$\cdot L^{-1}$ caused the fish to remain at the water surface. This behaviour was not observed in lower inorganic sediment strengths or in any concentration of organic sediment (Table 1).

Surfacing of fish in the test streams was not observed during the 6-wk study period. Other observations of behavioural responses (e.g. coughing, threats, nips, swimming activity) could not be made due to the opacity of the test suspensions.

Feeding-behavioural trials using live surface drift (*D. melanogaster*) showed that response times increased consistently with progressive increases in suspended sediment strengths. At 100 mg·L^{-1}, the response times varied from 16 ± 4 to 32 ± 15 s; at 300 mg·L^{-1}, from 24 ± 20 to 82 ± 37 s; and at 1000 mg·L^{-1}, from 25 ± 9 to >191 s. Response times for the control streams were similarly rapid (range 6 ± 1 to 8 ± 4 s).

In all but 3 of the 45 separate tests in the 1000 mg suspended sediment·L^{-1} streams, the fruit fly was consumed within the 360-s test period. Miss-strikes by grayling were noted frequently in 300 and 1000 mg suspended sediment·L^{-1} streams but not for those in control water or 100 mg suspended sediment·L^{-1}. Similar observations were noted during routine feeding, at which time fish in the higher sediment strengths initiated feeding later than those held in ≤100 mg suspended sediment·L^{-1}.

The feeding response of the naive *T. arcticus* (those held in streams for only 1 h prior to testing) to surface drift was more variable than that recorded for fish exposed for 5 wk to the test conditions. Response times for naive fish in control streams were 10 ± 5 to 25 ± 29 s; in 100 mg·L^{-1}, 10 to >360 s; in 300 mg·L^{-1}, 14 to >360 s; and in 1000 mg·L^{-1}, 209 to >360 s. As for the acclimated grayling, mean response times for these naive fish increased with sediment concentration but were relatively longer than those for the fish reared in test streams for 5 wk prior to evaluation. Failure to feed was noted for 18 of the 24 tests with naive fish in 1000 mg suspended sediment·L^{-1} and for 3 or 6, respectively, of the 24 tests carried out in 100 or 300 mg·L^{-1} sediment. Miss-strikes were again noted for fish held in the higher sediment strengths (Table 2).

The response of *T. arcticus* to subsurface prey (live brine shrimp) in three trials, after 6 wk of exposure, was similar for identical treatments. No consistent differences in response times to subsurface drift were found for groups of fish reared and tested in 0, 100, and 300 mg suspended sediment·L^{-1}. The mean response time for the consumption of all brine shrimp was 3–4 min in control streams, 3 min in 100 mg sediment·L^{-1}, 4–8 min in 300 mg sediment·L^{-1}, and >60 min in 1000 mg sediment·L^{-1} (the fish failed to consume all prey at this sediment strength within the 60-min test period) (Table 2).

Suspended sediment strengths of 100 and 300 mg·L^{-1} did not affect the feeding response times for *T. arcticus* held in test streams for 5 wk or 1 h and fed live tubificid worms. Regardless of these differing durations of sediment exposure, mean times for the consumption of all worms by control fish or those held in 100 or 300 mg sediment·L^{-1} were 3–10 min, and no consistent changes in response due to treatment were observed. However, the highest sediment strength examined (1000 mg·L^{-1}) reduced the feeding response of both naive fish and those exposed to this sediment concentration for 5 wk, with only 0–10% of the tubificids consumed in any test during the 60-min period (Table 2).

As with other salmonid fish species, the feeding habits of Arctic grayling vary depending upon life stage. Underyearling grayling feed primarily on zooplankton or drift (larvae) from benthic invertebrates whereas larger juveniles (>13 cm) or adult grayling tend to feed on benthic or emergent insects and larger terrestial insect drift (O'Brien et al. 1979; Stuart and Chislett 1979; Schmidt and O'Brien 1982; Birtwell et al. 1984; Simmons 1984). An examination of stomach contents for underyearling grayling captured during summer months from clearwater Yukon streams indicated that these fish were feeding principally on aquatic invertebrate drift (Chironomidae, Simuliidae) (Birtwell et al. 1984).

Schmidt and O'Brien (1982) determined that the reactive distance (i.e. distance within which a positive feeding response occurred) of Arctic grayling to a number of live zooplankton species was increased with increasing light intensity. Although several salmonid fish species appear to reach their maximum visual acuity at about 100 lx (Schmidt and O'Brien 1982), these investigators found that reactive distances for grayling increased up to a light intensity of 20×10^3 lx. Based on these findings, these authors concluded "Because grayling, at least in the Arctic, may do much of their feeding under conditions of continuous daylight and very clear water, (genetic) selection for low light vision may be low." Our findings appear to support the previous evidence that grayling rely on visual cues to locate insects and that decreased light intensity (due in the present instance to suspended solids) will impair feeding responses for this species.

Our results also indicate clearly that a 1000 mg·L^{-1} suspension of placer mining sediment markedly impairs the feeding performance of underyearling grayling offered surface drift, subsurface drift, or benthic invertebrates. This impaired response is evident for both naive fish and those exposed continuously to this suspended sediment strength for 5–6 wk. Interpretation of the effects noted for fish held in the lower sediment strengths (100 or 300 mg·L^{-1}) is less clear and confounded by modifications in the experimental approach used with differing food organisms. The increase in time to respond to surface drift for fish chronically exposed to 100 or 300 mg sediment·L^{-1} suggests that these sediment strengths (and 1000 mg·L^{-1}) decreased the fishes' reactive distance (ability to detect surface prey). The relatively slower response to surface drift for naive grayling held only briefly in these sediment strengths prior to testing may indicate that those fish subjected to prolonged sediment exposure have improved their ability to discern surface food in turbid water. Alternatively, short-term exposure to these sediment strengths may have disrupted feeding activities to some extent. The similarily rapid feeding response times for naive controls versus long-term controls show that the feeding activity of naive fish was not disrupted by transfer of fish from the stock tank to the test apparatus.

Unlike the findings with surface drift, suspended sediment strengths of 100 or 300 mg·L^{-1} did not impair feeding response times for grayling offered subsurface drift (*A. salina*) or benthic invertebrates (tubificids). The ability of naive or chronically exposed grayling to detect and consume tubificids or brine shrimp in these sediment strengths, as effectively as control fish, was likely due at least in part to the experimental design for this test. The white background provided by the submersed nylon mesh baskets may have silhouetted these prey, making their detection easier. Additionally, holding fish within the confines of each basket may have decreased the distance between predator and prey sufficiently for these fish to readily detect (and consume) these organisms at these sediment strengths. Or perhaps the reactive distance for these subsurface food organisms at each respective suspended sediment strength, whether related to visual or olfactory cues, is greater than that for surface drift.

Several investigators have reported a reduction in the feeding responses of salmonid fish due to the presence of suspended sediment. Noggle (1978) found that a suspended sediment strength of 100 mg·L^{-1} reduced feeding of coho salmon smolts toward caddisfly larvae by 45%, and that feeding ceased altogether above 300 mg·L^{-1}. Although both juvenile cutthroat trout (*Salmo clarki*) and chinook salmon can continue to feed on surface drift in suspended sediment concentrations >500 $mg \cdot L^{-1}$ (Griffin 1938), one study (Anonymous 1965) reported that resident cutthroat trout subjected for 2 h to 35 mg suspended sediment $\cdot L^{-1}$ sought cover and stopped feeding. Berg (1982) determined that suspended sediment at a turbidity level of 60 NTU had a marked effect on the visual ability of juvenile coho salmon. Delayed response times to surface drift, miss-strikes at food, and frequent collisions of fish with an obstacle within the test tank were evident. These findings are consistent with those for Arctic grayling (particularly naive fish) offered surface food in the presence of sediment suspensions $\geq 100\ mg \cdot L^{-1}$. Sigler et al. (1984) considered that the reduction in growth rate of coho salmon and steelhead trout during their exposure to suspended sediment was perhaps related to an inability to feed normally. This was confirmed by Simmons (1984) for Arctic grayling held under field conditions. He found that the feeding success of juvenile grayling was impaired in turbid waters. However, since prey availability was also reduced in this turbid environment (Wagener 1984), the reduced feeding success reported by Simmons (1984) may also (in this instance) have been a function of prey availability.

Fish Distribution

Based upon subjective observations, fish held in the higher sediment strengths ($\geq 300\ mg \cdot L^{-1}$) were usually distributed in the downstream half of the test streams during feeding. Unlike this situation, grayling held in the control and 100 mg sediment $\cdot L^{-1}$ streams were distributed along the length of each stream. This pattern of distribution was evident within 1 h of the introduction of sediment and thereafter throughout the 6-wk exposure. Quantitative studies of fish distribution after 4 and 5 wk of exposure showed that approximately 60% of the control fish occupied the upstream half of the control stream whereas only 32% of the fish exposed to 100 $mg \cdot L^{-1}$, 15% to 300 $mg \cdot L^{-1}$, and 16% to 100 $mg \cdot L^{-1}$ were found in the upstream half. These data, together with the subjective observations of fish distribution during regular feeding, indicate that the majority of fish were displaced downstream by the higher strengths of suspended sediment (300 and 1000 $mg \cdot L^{-1}$). Since the predominantly downstream distribution of grayling exposed to the higher suspended sediment strengths (300 and 1000 $mg \cdot L^{-1}$) was observed within 1 h of the initial establishment of the sediment gradients, it is unlikely that this behavioural response was caused by movement downstream in search of food. Rather, these findings suggest an innate downstream movement of these fish (avoidance?) in response to sediment exposure.

Sigler et al. (1984) found downstream displacement of steelhead trout and coho salmon fry from artificial streams receiving clay suspensions with turbidity values as low as 25 NTU. Noggle (1978) reported avoidance responses for juvenile coho salmon exposed to suspended sediment strengths of 4–8 $g \cdot L^{-1}$ whereas lower strengths (1–4 $g \cdot L^{-1}$) caused preference responses (fish attraction). Other investigators have reported no response (Gradall and Swenson 1982), preference (cited in Noggle 1978), or avoidance reactions (Anonymous 1965) for other species of salmonid fish exposed to low to medium ($<1000\ mg \cdot L^{-1}$) strengths of suspended sediment under controlled conditions. Berg (1982) determined that short-term pulses of suspended sediment with turbidity values ≤ 60 NTU caused a breakdown of social organization for juvenile coho salmon in laboratory stream environments, resulting in increased activity and a loss of aggressive interactions. From the foregoing, it is apparent that the behavioural responses of stream fish to suspended sediment and the associated characteristics of turbidity and cover are, as yet, unclear and that differences in fish species, age, and sediment strength and type (particle size and shape) may result in diverse behavioural reactions.

Fish distribution in natural stream environments can be markedly affected by suspended sediment loadings. Several instances of salmonid or other fish species avoiding muddy streamwater have been reported (Anonymous 1965). Herbert et al. (1961) reported an absence of brown trout (*Salmo trutta*) fry from downstream sites for streams receiving china-clay wastes whereas these fish were abundant at upstream, clear-water sites. Similarly, Birtwell et al. (1984), Pendray (1983), and Simmons (1984) found a consistent reduction in numbers of juvenile Arctic grayling within downstream waters receiving suspensions of placer mining sediment, relative to numbers found in upstream creekwater or clearwater tributary streams. These findings provide further evidence for the displacement of juvenile Arctic grayling or other salmonid fish species from stream environments due to high suspended sediment loadings.

General Discussion

The present laboratory bioassays demonstrate that under-yearling Arctic grayling can survive short-term exposure to very high levels ($\leq 50\ g \cdot L^{-1}$) of suspended inorganic or organic sediment and prolonged (6 wk) exposure to lesser ($\leq 1000\ mg \cdot L^{-1}$) concentrations of inorganic sediment. Acclimation temperature (and perhaps season) does not cause any marked change in this tolerance, although test results suggest a slight reduction in lethal tolerance to suspensions of inorganic sediment for grayling acclimated to cold (5°C) water. These laboratory findings are consistent with the survival data for grayling held for 4 or 5 d in turbid waters downstream of placer mining activity (Birtwell et al. 1984; Simmons 1984).

No gill histopathologies were found in these laboratory studies which could be attributed to exposure of grayling to sediment. Additionally, the laboratory bioassays indicated that the tolerance of grayling to hypoxic conditions or to upper lethal temperatures was not appreciably affected by suspensions of inorganic or organic sediment. The environmental significance of the slight but consistent decrease in critical thermal maxima for warmwater-acclimated grayling exposed to inorganic or organic sediment suspensions is not known at the present time. Nor is the significance of the increased time to death (reduced oxygen uptake rate) for warmwater-acclimated grayling only, caused by high ($\geq 4\ g \cdot L^{-1}$) concentations of inorganic sediment, understood. The decreased times to death for grayling held in sealed jars containing organic sediment concentrations $\geq 160\ mg \cdot L^{-1}$ are thought to reflect the oxygen demand of this organic material. The environmental relevance of this oxygen demand is site specific and would be modified markedly by factors such as overburden type and loading to receiving waters, flow conditions, water temperature, and presence or absence of ice cover.

The acute stress bioassays demonstrate that suspensions of both inorganic and organic sediment can be stressful to under-yearling Arctic grayling. Further, the test results indicate that suspended sediment strengths as low as 50 $mg \cdot L^{-1}$ (organic sediment) may be stressful to these fish and that stress responses can be evoked for both coldwater- and warmwater-acclimated fish. The environmental relevance of these responses to the immediate survival and long-term well being of Arctic grayling cannot be ascertained without further studies. However, stressful conditions are well known to reduce the adaptive responses of other salmonid fish species to natural environmental fluctuations and to increase the susceptibility of fish to disease (Wedemeyer et al. 1976, 1984; Wedemeyer and McLeay 1981). Sublethal effects including a continued impairment of feeding activity, impaired growth, decreased scope for activity, and decreased resistance to other environmental stressors can occur in response to stress.

Our findings indicate that concentrations of suspended placer mining sediment as low as 100 $mg \cdot L^{-1}$ can affect fish growth and feeding responses and that strengths $\geq$300 $mg \cdot L^{-1}$ can increase oxygen consumption (metabolic rate), lower the tolerance of grayling to a reference toxicant, and cause fish to be distributed further downstream. These findings provide cause for concern if sediment concentrations >100 $mg \cdot L^{-1}$ remain suspended in streams inhabited by juvenile Arctic grayling or other sensitive fish species for extended periods. Suspended sediment strengths as low as 100 $mg \cdot L^{-1}$ may also prove harmful to the long-term well being of grayling in natural stream environments. Although the effects on growth and feeding response times for fish exposed to 100 mg sediment$\cdot L^{-1}$ for 6 wk were minimal, the absence of a continuous supply of excess food in the natural environment together with the greater effort required for the detection and capture of available food, predator/prey interactions, and simultaneous exposure to other less than optimal environmental conditions may increase the impact of prolonged exposure to this suspended sediment strength.

The displacement of salmonid fish from upstream waters shown in this study for suspended sediment strengths >100 $mg \cdot L^{-1}$ and in a previous (Sigler et al. 1984) laboratory investigation is of particular importance, as these findings indicate that short-term pulses of suspended sediment may cause downstream migration of otherwise resident fish. This downstream displacement and probable reduction in the amount and quality of habitat are cause for concern if healthy natural populations of *T. arcticus* are to be maintained and managed in areas subjected to placer mining.

These and other studies indicate that a number of sediment characteristics besides concentation in suspension (i.e. particle size, shape, hardness, organic content) can modify their ability to harm salmonid or other fish species. Additionally, specific test conditions including water quality, fish handling and prior history, differences in fish condition, and innate tolerance could markedly alter the toxic responses of the same or other fish species to suspended sediment. Accordingly, the concentrations of organic or inorganic sediment shown in the present studies to cause specific responses are not applicable in a general sense, and values derived are useful only insofar as they demonstrate the nature and degree of effects that can be evoked.

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